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CASE STUDY

Accounting for transportation impacts in the environmental assessment of waste management plans

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Abstract
Background, aim, and scope Many recent studies on waste management have described in detail the potential impacts of recycling and final treatment of municipal waste. In public debates, the attention has also been focused on the choice of final disposal technologies (e.g. landfilling vs. incineration). However, a comprehensive assessment of the impacts of waste collection and transport was still lacking. In the present study, we use LCA to evaluate the potential impact of the provincial waste management plan of Varese (northern Italy). Particular attention is devoted to the estimation of environmental impacts generated during waste transport.
Materials and methods A detailed Life Cycle Inventory was built for the transportation phase, based on primary data collected by interviewing the agencies involved in waste collection. To model the recycling and final disposal phase we relied on the BUWAL 250 database. Impacts were evaluated with the Eco-Indicator 99 method in its egalitarian formulation.
Results The results of our analysis reveal that the major potential impacts of the plan are associated with waste collection and transport. These impacts are partially compensated by reduced resource consumption through recycling and energy recovery through incineration.
Discussion The outputs of the LCIA were compared with those obtained by using other ecoindicators (Eco-Indicator 99 hierarchist and individualist, CML2, EPS2000). Although not comparable on a quantitative basis, they are qualitatively consistent.
Conclusions Neglecting the effects of collection and transport might result in a severe underestimation of the environmental impacts of a waste management system, especially as refers to depletion of fossil fuels, emission of respiratory inorganics and climate change. To reduce the environmental impact of waste management systems, an accurate optimisation of waste transport is required.
Recommendations and perspectives Effective waste management planning requires the explicit inclusion of waste collection and transport when comparing alternative management policies.

Keywords Case studies · Life cycle assessment (LCA) · PPP impact assessment · Waste collection and transport · Waste management planning

Abbreviations

CML2	CML2 baseline 2000
EI99	Eco-Indicator 99
EI99e	Eco-Indicator 99 egalitarian
LCA	life cycle assessment
LCIA	life cycle impact assessment
MSW	municipal solid waste
PPPs	policies, plans and programmes
PWMP	provincial waste management plan
RFD	recycling or final disposal
WCT	waste collection and transport
WMS	waste management system

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1 Background, aim and scope

Life cycle assessment (LCA) was originally developed to analyse the life cycle impacts of products. During the last 10 years, however, its use has been extended to assessing potential environmental impacts of policies, plans and programmes (PPPs). In particular, the LCA approach has been applied to the assessment of waste management plans (Finnveden 1999), with the aim of dealing with waste in a way that is environmentally efficient, economically affordable and socially acceptable (Barton et al. 1996; Clift et al. 2000; Thomas and McDougall 2005). A key feature of LCA is that all products or services being compared must provide the same functions, so that they can be compared on equal grounds; this aspect is essential when comparing different waste management strategies, which may provide different outputs, such as recovered materials or energy. Sustainable waste management requires an environmentally oriented approach that can be pursued through the development of waste management systems (WMS) (Beccali et al. 2001). A WMS can be subdivided into two main steps:

1. Waste collection and transport (WCT);
2. Recycling and final disposal (RFD).

Comparisons of different waste management strategies through LCA have mostly focused on RFD. The impact of WCT was analysed in detail only recently, and in most cases its contribution was considered to be negligible (Eriksson et al. 2005). Bruno et al. (2002) assigned a marginal role to potential impacts of the WCT step when comparing different WMSs for the Cuneo province (Northern Italy). Finnveden et al. (2005) compared different WMSs in Sweden by considering different transportation scenarios, but could not impute any significant difference among treatment strategies to the impact of transportation. In contrast, Bergsdal et al. (2005), by comparing localised and centralised incineration strategies in Norway, showed that WCT impacts are comparable to those caused by incineration and suggested designing WMS so as to minimise travelled distances. Beigl and Salhofer (2004) compared kerbside and bring collection systems in Austria, concluding that individual transport has a stronger impact compared to collective WCT. Yet, a comprehensive comparison between WCT and other WMS impacts was lacking in their work. Muñoz et al. (2004) considered WCT impacts in their analysis of a 700,000 people district in Spain. Although the collection step was recognised to be the most impacting one, they did not find significant differences between alternative scenarios with a different number and different types of treatment plants. In a recent paper, Salhofer et al. (2007) investigated the environmental relevance of transport in disposal systems of specific waste streams in Austria by comparing recycling with longer transport distances with disposal without recycling. Four material streams were considered: refrigerators, waste paper, polyethylene films and expanded polystyrene. Despite longer transport distances, recycling was environmentally more beneficial than disposal. However, their study provided evidence that the impact of waste transport cannot be neglected. Finally, Björklund and Finnveden (2005) recently reviewed a large number of LCA studies on WMS in Europe, and none of them considered WCT as a source of potentially considerable environmental impacts.

2 Methods, goal and scope

Waste management in Italy has been characterised by a long period of deregulation, with landfill disposal as the main treatment method, without any form of energy recovery or material selection (a few incinerators were present, mainly in the north of the country, but often with obsolete technologies). The first important improvement was obtained in 1997 with the so-called 'Decreto Ronchi' (Italian Legislative Decree N.22, 5 February 1997) that adopted the European legislation and introduced three basic concepts for the management of municipal solid waste (MSW): reusing, recycling and energy recovery. Waste management planning is delegated to provinces through the development of provincial waste management plans (PWMPs). The Varese province (northern Italy), a district of approximately 840,000 inhabitants covering about 1,200 km^2 (Fig. 1) and producing about 400,000 t of MSW each year (OPR Varese 2005), recently approved a PWPM to provide a framework for sustainable waste management (Provincia di Varese 2005). As for source separation, Varese is one of the provinces in Italy performing most extensively, with 51% of the waste sent to recycling facilities in 2005 (OPR Varese 2005; compared to 41% in northern Italy and 23% in the whole country; APAT/ONR 2006). Nonetheless, the Varese PWMP lacks a systematic analysis of the environmental effects of the WCT step and does not consider possible strategies for the reduction of WCT impacts.

The aim of this work was to assess the Varese PWMP by (1) quantifying the potential environmental impacts of the WCT step and (2) comparing them with those caused by recycling and final disposal of MSW. LCA can be effectively used to comprehensively assess the potential impacts of a WMS, pointing out the most impacting phases and suggesting which processes can be adjusted by decision-makers to improve environmental performances. According to ISO (2006) guidelines, LCA must evaluate the potential impacts of a product throughout the whole product life (cradle-to-grave). When this technique is used to assess WMSs, however, system boundaries are to be

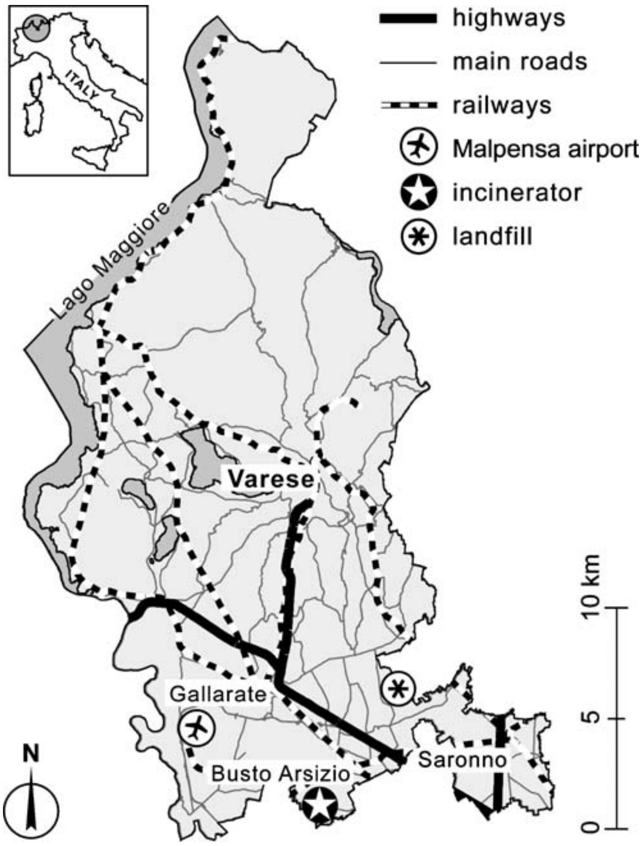

Fig. 1 The Varese province

reshaped in order to include only processes starting when materials become waste, without considering the potential impacts generated before this point (Consonni et al. 2005). Accordingly, our model starts from waste collection and ends with final recycling or disposal. The attention was specifically focused on the impacts of the WCT phase.

First, we developed a waste transport model (Section 3) to analyse WCT in detail. Then, we performed the LCIA using Eco-Indicator 99 (Section 4; EI99 in the following). Finally, we assessed the sensitivity of the results with respect to the weighting set used to aggregate impacts and compared results obtained with EI99 with those obtained with two other ecoindicators commonly used in European LCA studies (Section 5). This comparison is useful to evaluate to which extent the choice of a particular ecoindicator influences final outcomes.

The functional unit is 1 t of MSW collected and treated in the Varese province. Primary data on waste production and collection are referred to 2003 (OPR Varese 2004). At that time, total MSW production in the province was 398,460 t. Of these, 46% was source separated and sent to recycling or composting, while the remaining 54% was disposed of as mixed waste. The geographical area under investigation corresponds to the collection area of MSW. According to ISO 14040 standards, system boundaries were extended outside the province to include recycling treatments (Ekvall and Finnveden 2001). At present, the only waste treatment plants of the Varese province are indeed a landfill and an incinerator, located in the southernmost part of the district.

The LCIA was performed with the SimaPro 6 software (Goedkoop and Oele 2004). This software utilises the ecoindicator approach. After building the Life Cycle Inventory (LCI), a set of impact indicators is associated with each elementary action. Then, scores are summed up along the whole life cycle to obtain a global score for each impact category. Finally, global scores can be aggregated into a single number (the so-called ecopoint score) by means of a weighted sum. The choice of weights reflects the viewpoint of the decision-maker, and is therefore a subjective step. For this reason, ISO 14040 guidelines do not allow the use of aggregated results in public comparisons. Among the several ecoindicators currently available, we chose Eco-Indicator 99 (Goedkoop and Spriensma 2001) which has been widely used in a number of Italian and European LCA analyses. EI99 disaggregates environmental effects into 11 impact categories (namely carcinogens, respiratory organics, respiratory inorganics, climate change, radiation, ozone layer, ecotoxicity, acidification and eutrophication, land use, minerals, fossil fuels), that in turn are grouped under three macro categories (human health, ecosystem quality and resources depletion). Three weighting sets have been proposed to aggregate EI99 indicators: ‘hierarchist’, ‘egalitarian’ and ‘individualist’ (in the following EI99h, EI99e and EI99i, respectively), reflecting three different social perspectives based on the cultural theory of risk (Thompson et al. 1990). The individualist perspective has a short-term horizon and scarce interest in low-probability impacts. The hierarchic one balances short and long-term horizons and has a consensus-based approach to risk. The egalitarian one has a long-term horizon and relies heavily on the precautionary principle. Using different weighting sets may lead to very different final impact estimates. In this work, we adopted the egalitarian weighting set as the primary indicator. Because of its long-term perspective, we consider it to be the most appropriate for the assessment of PPPs.

3 Inventory

3.1 Waste collection and transport

WCT can be divided into two steps: (1) waste collection within municipalities and (2) transport from municipalities to landfills, incinerators or recycling plants. Waste is first collected by waste collection vehicles or vans and convoyed to waste sorting facilities. Then, source separated waste is transported by trucks to disposal and recycling

Table 1 Vehicle types for WCT

Vehicle	Max. load (t)	Units	Use
Waste collection vehicle	n.a.	h	Urban MSW collection and compacting
Delivery van	3.5	t km	Urban door-to-door MSW collection
Truck	16	t km	Long-distance transport

plants. We approximated the vehicle set actually used in Varese with three vehicle types available within the BUWAL 250 database having similar characteristics (see Table 1 for details). Emissions by waste collection vehicles depend primarily upon the duration of waste compacting and are proportional to operating time. Emissions by vans and trucks are proportional to the product of the load charged (in tons) and the distance covered (in kilometres).

Not all the municipalities use the same collection method. Each municipality can indeed decide upon the materials to be selected and the way to collect them. Two main collection methods can be distinguished: door-to-door and recycling bank collection. A second categorisation can be performed on the basis of population density. In fact, the population of the Varese province is unevenly distributed over the territory. On average, population density is quite high (ca. 700 inhabitants/km^2). However, the southern part of the province is flat and has a very high density (more than 1,200 inhabitants/km^2), while the northern part, which is hilly or mountainous, is characterised by large empty spaces, small villages and a very low density. As the distance travelled to collect a ton of MSW should typically be shorter in municipalities with higher population density, we subdivided municipalities into two categories to obtain a more precise estimate of travel distances: high density (>500 inhabitants/km^2) and low to medium density (with ≤500 inhabitants/km^2). We approximated the operating time of collection vehicles with the duration of working shifts. These last approximately 6 h in high density municipalities and 2 h in low density municipalities. Data on collection methods, waste loads and travel distances were acquired by interviewing the three most important collection companies of the province, covering more than 80% of the province population. Data gaps were filled by using average data derived from a national study (ANPA 1999). Waste amounts transported by collection companies in 2003 are summarised in Table 2. Travelled distances by vehicle type are reported in Table 3. For instance, a ton of plastics is (on average) first transported by a collection vehicle for 3.20 h; then, it is transferred for 117.90 km by Van; finally, it travels 27.02 km more on truck to recycling plants. Some waste types (such as green, wood, metals, bulky waste) are directly transported by citizens to civic amenity sites. For these, only the transport from civic amenity sites to final disposal facilities was accounted for in the analysis. Travel distances are in some cases very long (in particular for mixed waste), as disposal plants are sometimes outside the province boundaries. By aggregating data for municipalities with low/medium and high density we built the global WCT inventory for the whole province.

3.2 Recycling or final disposal

Where source separation is performed, the final destination of MSW depends upon waste type. Final treatments include:

- Recycling (paper, glass, plastics, metals);

Table 2 Waste collected within municipalities, by source separated waste type, collection method and population density

Waste type	HPD door-to-door (t)	HPD recycling bank (t)	LPD door to door (t)	LPD recycling bank (t)	Total (t)
Paper	29,607	1,305	4,783	1,573	37,268
Glass	19,121	8,385	4,697	2,600	34,802
Plastics	7,482	1,070	1,851	350	10,753
Organic	23,968	0	2,486	0	26,454
Mixed	136,266	0	32,807	0	169,073
Green					40,261
Wood					13,400
Metals					9407
Bulky waste					36,218
Other					14,279

HPD High population density; *LPD* Low/medium population density. Green, wood, metals, and bulky waste are directly transported by citizens to civic amenity sites. Totals do not sum up exactly to 398,460 t because ca. 6000 t of waste, whose ultimate fate was not known, were not included in the analysis

Table 3 Average vehicle usage for collection and transport of 1 t of different waste types

Vehicle	Unit	Paper	Glass	Plastics	Organic	Mixed	Green	Wood	Metals	Bulky waste	Other
Waste coll. vehicle	(h/t)	1.45	0.00	3.20	0.00	0.87	0.00	0.00	0.00	0.00	0.77
Van	(km)	113.30	26.51	117.90	140.34	30.77	0.00	0.00	42.62	15.15	36.67
Truck	(km)	58.47	90.90	27.02	80.00	265.05	145.79	40.61	0.00	0.00	20.00

- Composting (organic, wood and green waste);
- Disposal in incinerator or landfill (bulky, mixed and other non-separated waste).

The totality of waste amounts (see Table 2) separated for recycling or composting is transported to the relevant treatment plants. Sixty-five percent of bulky, mixed and other waste is disposed of in the landfill, while the remaining 35% is sent for incineration.

Comprehensive primary data on the operating inputs of waste treatment facilities were not available. Hence, we accounted for the environmental impacts of all the processes considered, encompassing secondary raw material production by referring to the average characteristics of European plants with the same technical features and a similar geographical context from the BUWAL 250 database. To assess environmental benefits of recycling, we accounted for the savings of virgin materials (pulp for cardboard, glass, polyethylene as a mixture of HDPE and LDPE, and iron) which are substituted by secondary raw materials obtained through recycling (paper, glass, plastics and metals, respectively).

The biogas produced by the landfill is captured and used to produce electricity (4,622 MWh/year, corresponding to ca. 37 kWh per ton of MSW). The incinerator is also used for electricity production, while it is not used for heat production at the moment (although the development of a district heating network is expected on a mid-term horizon). To quantify the environmental benefit of energy recovery through biogas production and incineration, we have calculated the (avoided) impacts to produce an equivalent energy amount with the average generation mix of the European (UCTE) grid.

4 Results of the impact assessment

Results of the LCIA as obtained with EI99e are shown in Fig. 2. Positive impacts indicate an environmental impact (emission or energy use), whilst negative ones indicate an environmental benefit (emission reduction or energy saving). Figure 2a shows the potential impacts of WCT by waste type, grouped under the three macro categories. The major impact is related to resource depletion. The most impacting process is collection and transport of mixed waste

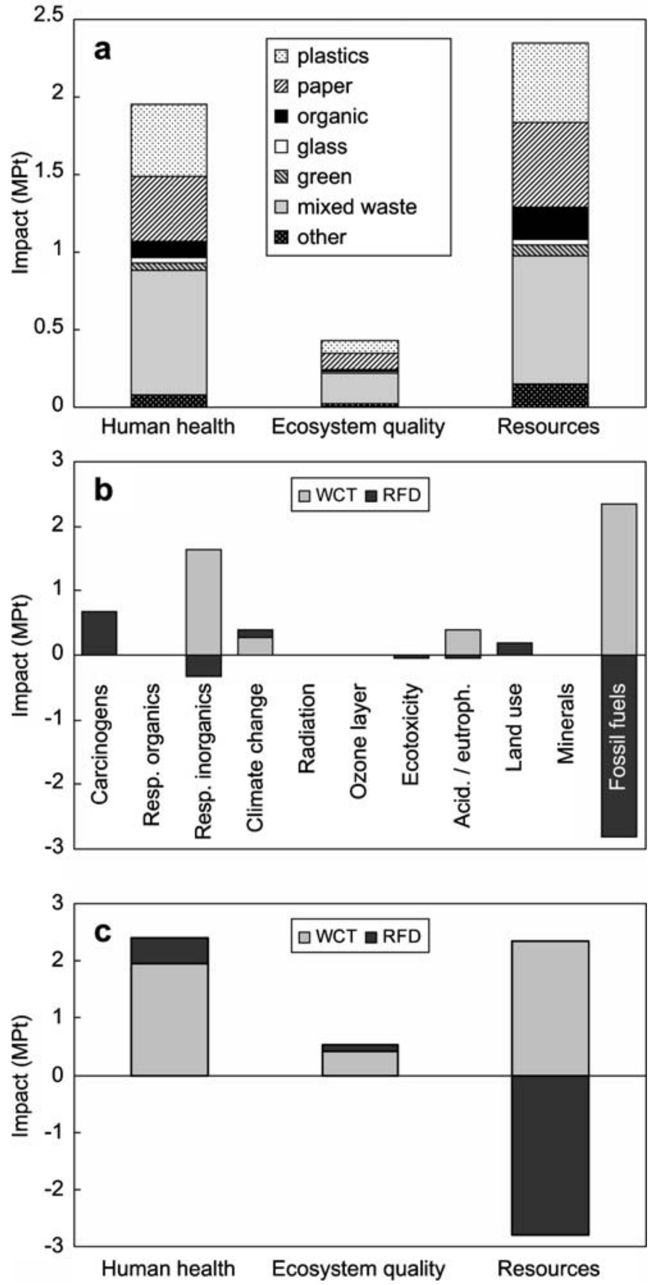

Fig. 2 Impacts of the Varese PWMP evaluated with the EI99 ecoindicator (egalitarian). **a** WCT step, by source separated waste type, grouped under 3 impact macro categories; **b** whole WMS, grouped under II impact categories; **c** whole WMS, grouped under 3 impact macro categories

Table 4 Impacts generated by different waste types evaluated with the EI99 egalitarian ecoindicator

Waste type	WCT (Pt/t)	RFD (Pt/t)	Total (Pt/t)	PWMP (MPt)
Paper	27.6	−6.6	21.0	0.78
Glass	3.2	10.3	13.6	0.47
Plastics	100.4	−179.5	−79.0	−0.85
Organic	12.9	2.9	15.8	0.42
Mixed	10.6	3.5	14.1	2.39
Green	3.6	2.9	6.5	0.26
Wood	1.4	10.7	12.2	0.16
Metals	2.3	−144.6	−142.2	−1.34
Bulky waste	0.4	0.0	0.4	0.01
Other	12.4	0.0	12.4	0.18

(40% of the impact on human health, 43% on ecosystem quality and 35% on resources). Plastic transport also has a quite high impact (24%, 21% and 23% of the impact on each macro category, respectively). Due to the low density of plastic waste, a large number of trucks are indeed needed to collect it (Muñoz et al. 2004). Major emissions, in terms of ecopoints score, come from mixed waste (38%), plastics (23%), paper (22%), organic (7%), green (3%) and glass (2%), while all other waste types contribute to less than 1%. If WCT impacts are subdivided by vehicle type, the largest contribution is that of waste collection vehicles (51%), which are mainly used in door-to-door collection of mixed waste, and, in part, to collect plastics and paper. The relative impact of delivery vans and trucks for long-distance transport is similar and about one half that of waste collection vehicles (21% and 28%, respectively).

Figure 2b compares the effects of the two WMS steps (WCT and RFD), disaggregated into the 11 EI99 impact categories. WCT obviously generates only positive impact values, because all processes produce emissions. In particular, most impacts stem from diesel collection trucks operating with changing load. RFD generates both positive and negative impacts on the environment. Negative ones stem from final disposal through incineration (which produces electric energy) and recycling processes (which reduce the consumption of virgin materials). The main potential impacts of the whole WMS are related to the consumption of fossil fuels and the release of inorganic compounds with respiratory effects. Both impacts are largely imputable to truck emissions. Relevant impacts are also connected to carcinogens, climate change and acidification/eutrophication. Except for carcinogens, the major impacts of the WMS are imputable to WCT, while RFD generates negative impacts on most impact categories (and in particular on fossil fuels and respiratory inorganics). The overall environmental impacts of the two steps, disaggregated by waste type, are compared in Table 4. The major environmental benefits come, in the order, from recycling plastics, metals and paper. Recycling glass produces, on the whole, no environmental discounts, as material savings are overwhelmed by electricity usage.

In conclusion, the highest environmental benefit is that related to recycling of plastics, while the highest environmental load is associated with the transport of mixed waste. The management of metal waste has the most favourable environmental balance, with a net benefit of 1.34 MPt. While the impacts on climate change and fossil fuel depletion have the same sign for WCT (the more fossil fuels are burnt for waste transport, the more greenhouse gases are released into the atmosphere), the same does not hold for RFD. The major cause of this counterintuitive effect is the recycling of plastics. In fact, fossil fuels are used in plastic production not only as an energy source, but also as a raw material. In this second case, fossil fuels have almost no impact on climate, as they are immobilised within the final product. Therefore, recycling plastics allows a considerable saving of fossil fuels as raw materials, but not for energy production, and this is the reason why the RFD impact on climate change is positive.

Potential impacts generated by the whole WMS, split into the three macro categories cited before, are illustrated in Fig. 2c. The impact on resource depletion is negative, while both WCT and RFD steps generate positive impacts on human health and ecosystem quality (with RFD accounting for about 25% of the total score). According to our analysis, the implementation of the Varese PWMP would completely compensate resource consumption caused by WCT through the recovery of energy and raw materials in the RFD step. On the contrary, both steps would determine positive impacts on both human health and ecosystem quality.

5 Comparison among ecoindicators

To assess the robustness of the results with respect to the choice of the ecoindicator, we repeated the LCIA by using

Table 5 Overall impact of WCT and RFD as estimated with the three EI99 weighting sets and the EPS2000 ecoindicator

Ecoindicator	WCT (MPt)	RFD (MPt)
EI99 egalitarian	4.72	−2.23
EI99 hierarchist	6.58	−3.45
EI99 individualist	2.30	1.18
EPS2000	25.1	−24.6

Note that scores are not comparable between ecoindicators, as the underlying metrics are different

the two alternative weighting sets available for EI99 (EI99h and EI99i, respectively) and two other ecoindicators, namely EPS2000 and CML2 baseline 2000 (CML2 in the following).

Table 5 compares generated and avoided impacts of the whole WMS (WCT+RFD) as obtained assuming the egalitarian, hierarchist and individualist perspectives. Impact estimates show slight differences between EI99e and EI99h, while scores obtained by using EI99i are considerably different, in particular with respect to the sign of the RFD impact (negative with EI99e and EI99h and positive with EI99i). An analysis of the two steps separately indicates that WCT impact estimates are not appreciably affected by the EI99 weighting set used, neither if aggregated on the whole WCT step nor when split into selected waste types. Although overall ecopoint scores are quite different in absolute terms, the relative contributions of different waste types to the overall impact of WCT are practically identical if assessed with EI99e or EI99h, while minor changes (in the order of 1–3%) are found by applying the EI99i weighting set. In all cases, mixed waste collection and the relevant vehicle usage are the most important emission sources. On the contrary, the assessment of RFD impacts is considerably affected by the choice of the weighting set. Results obtained with EI99i show indeed no environmental benefits of recycling plastics and glass, as electricity usage overwhelms the negative impact of material savings. This is not surprising, as EI99i does not account for fossil fuel depletion, whereas the other two weighting sets identify this as a key feature.

As regards the results obtained with EPS2000 and CML2, it should be noted that a comparison among different ecoindicators cannot be done on a strictly quantitative basis, but should be limited to a qualitative evaluation. The impact categories considered are indeed different. The EPS2000 ecoindicator (Steen 1999) was specifically designed to analyse and improve industrial processes. In its formulation, it is quite similar to EI99, but it uses four aggregated impact categories (human health, ecosystem production capacity, abiotic stock resource, biodiversity) instead of three as EI99 does. Results obtained with EPS2000 are

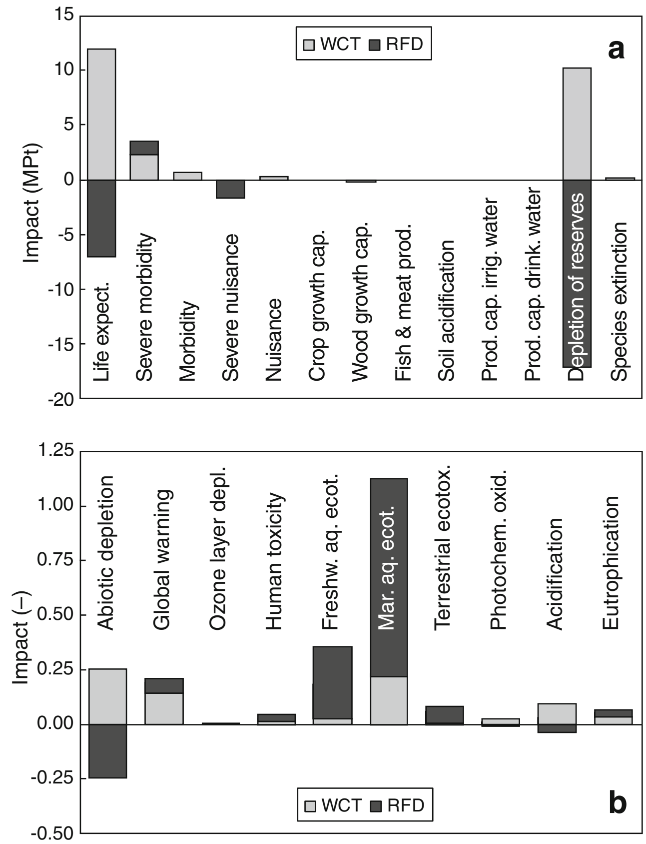

Fig. 3 Impacts of the Varese PWMP evaluated with **a** EPS2000 and **b** CML2 ecoindicators

shown in Fig. 3a and Table 5. The categories affected by the most severe impacts are life expectancy and depletion of reserves, both generated by the WCT step. The same impact categories are also characterised by considerable avoided impacts, to be imputed to the RFD phase. Results obtained with the EPS2000 ecoindicator are fully consistent with those obtained with EI99e, even if they are disaggregated by selected waste type: the major impacts are indeed those related to human health and resource depletion, and have a positive sign for WCT and a negative one for RFD.

CML2 (Guinée 2002), which calculates normalised impacts with respect to the 2000 European situation, considers ten impact categories and does not contemplate weighting. Therefore, impacts assessed with CML2 (Fig. 3b) are not comparable across impact categories and cannot be summed up to assess the overall impact of the two WMS steps. In contrast, it is possible to compare the relative importance of the two steps on the same impact category. The impact balance on abiotic depletion is very similar to that on fossil fuels estimated by EI99e (see Fig. 2a), with the positive impact of WCT approximately set off by the negative impact of RFD. Both EI99e and CML2 show that WCT and RFD have a positive impact on health

and ecosystem quality. However, the relative importance of the two steps is reversed when assessed with CML2 aquatic and terrestrial ecotoxicity categories. With respect to these categories, the major impact sources are leakage from the landfill and energy consumption for plastic recycling.

6 Conclusions

We assessed the potential impact of the Varese PWMP with the LCA methodology, focusing in particular on the contribution of waste collection and transport. WCT impacts were assessed with a comprehensive primary dataset acquired directly from collection companies operating in the province. The estimation of RFD impacts was based on secondary data taken from the BUWAL 250 library of SimaPro, which relies on a rather old database. We are aware that this discrepancy might have affected our results. However, plants and processes were suitably selected to provide the best approximation of those actually used in Varese, and we are confident that the overall dataset is acceptably time- and space-consistent.

Results clearly show that WCT is the most impacting step in all three macro categories considered in EI99. The impact of WCT on human health and ecosystem quality is around four times that caused by RFD. The impact on resources depletion is balanced off by the environmental benefit of RFD. About half of the overall environmental impact of WCT is compensated for by recycling and energy recovery through the RFD step. The most impacting processes are collection and transport of mixed waste, plastics and paper. As for plastics, results are consistent with what observed in the transport of light materials, such as expanded polystyrene (Salhofer et al. 2007). The estimated impact of paper is higher than observed in other studies (Muñoz et al. 2004; Salhofer et al. 2007), as the environmental impact of transport is only partly compensated by recycling. In the case of glass, recycling is an energy intensive process requiring so much energy that recycling impacts overwhelm environmental benefits. Notice, however, that incinerating or landfilling glass would determine even higher impacts.

The Italian legislation does not set specific geographical boundaries for waste disposal. Therefore, WCT systems often ensue from a series of individual local choices based on economical considerations rather than being the result of an optimisation process aimed at minimising travel distances. An exemplary case is that of Varese (the province capital), producing about 23,000 t of mixed waste per year (OPR Varese 2004). This waste is not disposed of within the province, but is transported to the Dalmine incinerator, near Bergamo, about 96 km away. This results in more than 3,300 round trips from Varese to Dalmine and viceversa, approximately corresponding to 640,000 km travelled by truck each year.

The results obtained with the egalitarian version of the EI99 ecoindicator were compared with those obtained with alternative weighting sets for EI99 (EI99h and EI99i) and two other ecoindicators (CML2 and EPS2000), with the aim of assessing the dependence of the results upon the choice of the ecoindicator. Although a quantitative evaluation was not possible, due to the different impact categorization used in different ecoindicators, a qualitative comparison of the scores obtained in analogous impact categories showed that estimated environmental impacts are consistent among ecoindicators.

Our results are drawn from the analysis of a specific study case, the PWMP of the Varese district. However, we believe that similar results might be obtained by applying the LCA approach to other Italian and, most likely, also European contexts with similar characteristics (waste management characterised by a fairly high level of waste selection and long travelled distances). Effective waste management planning requires to explicitly account for the impact of WCT when comparing alternative management policies. Public debates have generally focused on the choice of final disposal technologies (e.g. landfilling vs. incineration), but have usually ignored the collection and transport phase. However, neglecting WCT effects might result in a severe underestimation of the environmental impacts of a WMS, especially as refers to depletion of fossil fuels, emission of respiratory inorganics and climate change.

7 Recommendations and perspectives

Support methods for public decision-making, such as Strategic Environmental Assessment, can include LCA and other techniques with the aim of facilitating and rationalising a systematic and comprehensive analysis of the potential environmental impacts of PPPs. Further research is needed to gain a deeper insight into the impacts of waste collection systems and waste transport to its final destinations. Effective waste management must cope with conflicting objectives: on one hand, maximising source separation and recycling; on the other hand, minimising travel distances to minimise the impacts of waste transport at both the local scale (respiratory effects) and the global one (depletion of non-renewable resources and climate change). In this perspective, multi-objective techniques can provide decision-makers with a conceptual framework to tackle this trade-off.

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